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7. Abstract

Travel Time Estimates for Alternative Tritium Crib Locations Hanford Site, Washington was prepared by Golder Associates, Inc., for Westinghouse Hanford Company.

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1. INTRODUCTION

This report provides estimates of groundwater travel times to the Columbia River from eight alternative tritium crib locations on the Hanford Site. The estimates were made using a two-dimensional, finite element model of the uppermost aquifer at the Hanford Site. This model was prepared by Golder Associates to support an earlier investigation of alternative soil column disposal locations for process waste streams from the 200 Areas. This earlier investigation is summarized in Appendix H of the report "200 Area Treated Effluent Disposal Study" (Engineering-Science, Inc., 1989), and has been included for reference as Appendix A to this report.

This report is presented in four sections. Following this introduction, a discussion of the groundwater model is presented in Section 2. The input parameters for this study and the results obtained are discussed in Section 3, and conclusions and recommendations are presented in Section 4.

2. HANFORD SITE GROUNDWATER MODEL

The Hanford Site groundwater model used in this study was developed using Golder Associate's Golder Groundwater Package. This modeling package contains state-of-the-art finite element computer programs for simulation of groundwater flow and contaminant transport, as well as graphics programs for presenting results. For this study, the program AFPM (Aquifer Flow in Porous Media) was used in its two-dimensional form. The program accommodates variable aquifer properties, a changing phreatic surface, transient boundary conditions, and other characteristics useful for groundwater modeling at the Hanford Site.

Development and calibration of the groundwater model is discussed in detail in Appendix A, and will only be summarized here. The modeled region and finite element mesh are shown in Figure 1, and were determined based upon the principal geologic heterogeneities, groundwater flow patterns, and boundary conditions at the Hanford Site. The model contains 976 nodes and 920 elements. Most of the elements are square with side lengths of 3,275 ft.

Boundary conditions were defined as explained in Appendix A and shown in Figure 1. The base of the aquifer was estimated from Plate III-2 in Gephart et al. (1979). The thickness of the aquifer and therefore the transmissivity varied within regions of constant hydraulic conductivity. Initial hydraulic conductivity values were estimated from Plate III-5 in Gephart et al. (1979). These conductivities were then modified in a series of calibration runs until reasonably close comparisons were obtained for both 1944 and 1979 phreatic surfaces. The final hydraulic conductivities used in the model are shown in Figure 2. Effective porosities were determined using the model-generated flowpaths and the actual travel times of known tritium plumes on the Hanford Site. This process is also described in Appendix A. Effective porosities were found to be correlated with hydraulic conductivity, and are estimated to range from 0.15 to 0.25 as shown in Figure 2. The hydrogeologic properties and boundary conditions used in this study are the same as those developed for the aforementioned 200 Area study (Engineering-Science, Inc., 1989).

3. ALTERNATIVE CRIB ANALYSIS AND RESULTS

3.1 Model Parameters

Travel time analyses were made for hydrologic conditions on the Hanford Site with B-Pond in operation (Case 1), and without B-Pond in operation (Case 2). Both cases were studied using hydrologic boundary conditions developed for the aforementioned 1979 model calibration. In both cases, natural groundwater recharge was assumed to be provided only from subsurface inflow from Cold Creek and Dry Creek Valleys; no recharge was assumed from infiltration of direct precipitation. In the Case 1 study, additional artificial recharge was assumed only from B-Pond, and in the Case 2 study no additional source of artificial recharge was assumed. The results are therefore intended to represent near-future conditions when artificial recharge from all major facilities (except B-Pond in Case 1) has ceased and the underlying groundwater mounds have dissipated.

The hydraulic head contours for the uppermost aquifer under Case 1 conditions (with B-Pond) are shown in Figure 3. The groundwater mound beneath B-Pond is evident in the central part of the model area. The steady-state B-Pond inflow was assumed to be 16.5 million gallons per day (2.2×10^6 ft^3/day), based on information provided by Westinghouse Hanford Company (WHC).

The hydraulic head contours for the uppermost aquifer under Case 2 conditions (without B-Pond) are shown in Figure 4. This is the same as Figure H-7 of Appendix A.

3.2 Travel Time Results

Travel times were estimated for the eight crib locations A through H shown in Figures 5 and 6. Locations A through F were described in the initial WHC Task Order, and locations G and H were added from subsequent discussions with the WHC technical liaison. Inflow into the tritium cribs was assumed to be the same at each location, and equal to 11,300 ft^3/day. This inflow rate is sufficiently small that no mounding beneath any of the cribs could be discerned from the hydraulic head contour maps. The crib discharges were therefore assumed to have no influence on groundwater flow rates and directions.

The crib inflow is equal to the combined average flow of the effluent waste streams from PUREX Process Condensate (8,000 ft^3/day) and from the 242-A Evaporator Process Condensate (3,300 ft^3/day)(Engineering-Science, Inc., 1989, Table 2.1). These two waste streams have the highest average tritium concentrations and when combined account for approximately 88 percent of the total tritium release from the PUREX and 242-A Evaporator facilities (Engineering-Science, Inc., 1989, Table A.1).

Tritium travel time is expected to be the same as groundwater travel time because the tritium molecule is very similar to the natural water molecule and is non-sorbing. Estimated travel times are shown in Figure 5 for Case 1 with B-Pond, and in Figure 6 for Case 2 without B-Pond. All travel time results are summarized in Table 1. All travel times are expressed to the nearest whole year, without further rounding, to indicate the relative differences for the various crib locations; however, this should not be taken as an indication of the accuracy of the estimates. Although the actual accuracy of the estimates is not known, based upon a comparison of simulated results with observed plume travel times, the error may be approximated to be about plus or minus 30 percent.

Tritium travel times were found to range from 20 to over 130 years for the various crib locations and hydrologic conditions. In general, travel times without B-Pond are longer than with B-Pond because of the increased hydraulic gradients caused by the B-Pond mound. As would be expected, these differences are greatest for the locations near B-Pond. Exceptions occur only at crib locations A and H: the path length from crib A to the river is shorter without B-Pond and requires less travel time; and the path from crib H to the river passes through higher conductivity materials without B-Pond and requires less travel time.

Tritium crib H is of potentially greatest interest because of its estimated travel time in excess of 100 years. The travel time from this crib is large because of the long flow path within the zone of lowest hydraulic conductivity in the model, shown in Figure 2. About 70 percent of the travel time from crib H occurs within this low conductivity zone, and its influence on the results is therefore significant.

4. CONCLUSIONS AND RECOMMENDATIONS

Based upon the results obtained, the longest estimated travel time and therefore the most attractive location for a tritium disposal crib is at point H on Figures 5 and 6. Considering that the half-life of tritium is about 12.3 years, a travel time of 130 years would consume more than 10 half lives. The residual tritium concentration upon release to the river would be about 0.1 percent of the original concentration discharged to the crib.

Before making a final selection of crib location, a relatively simple sensitivity study of the model to the various uncertainties in input parameters is recommended. Of particular importance would be a thorough review of available hydraulic conductivity data and an evaluation of the effect of small amounts of groundwater recharge from direct precipitation on the calibrated hydraulic conductivity values. The uncertainties related to the primary mechanisms of groundwater recharge assumed in the model are discussed in Appendix A. The average rate of recharge on the Hanford Site from natural precipitation is currently the subject of extensive research by WHC and Pacific Northwest Laboratory personnel, and highly variable results have been obtained based on ground surface conditions (Gee, 1987, p. 5.1). The calibrated hydraulic conductivity values may change if recharge from direct precipitation is considered, and, as has been seen, the estimated travel time is relatively sensitive to hydraulic conductivity.

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Engineering-Science, Inc., 1989. 200 Area Treated Effluent Disposal Study. Prepared for Westinghouse Hanford Company, Richland, Washington, by Engineering-Science, Inc., Denver, Colorado, under contract to Golder Associates Inc., Redmond, Washington, April.

Gee, G.W., 1987. Recharge at the Hanford Site: Status Report. Pacific Northwest Laboratory Report PNL-6403, Richland, Washington, November.

Gephart, R.E., R.C. Arnett, R.G. Baca, L.S. Leonhart, and F.A. Spane, Jr., 1979. Hydrologic Studies within the Columbia Plateau, Washington: An Integration of Current Knowledge. Rockwell Hanford Operations Report RHO-BWI-ST-5, Richland, Washington.

TABLE 1

TRITIUM TRAVEL TIME RESULTS

Tritium Crib	Estimated Travel Time*	
	With B-Pond (years)	Without B-Pond (years)
A	22	20
B	35	54
C	16	48
D	56	72
E	86	87
F	64	69
G	75	80
H	134	126

* Estimated standard error is plus or minus 20 percent.

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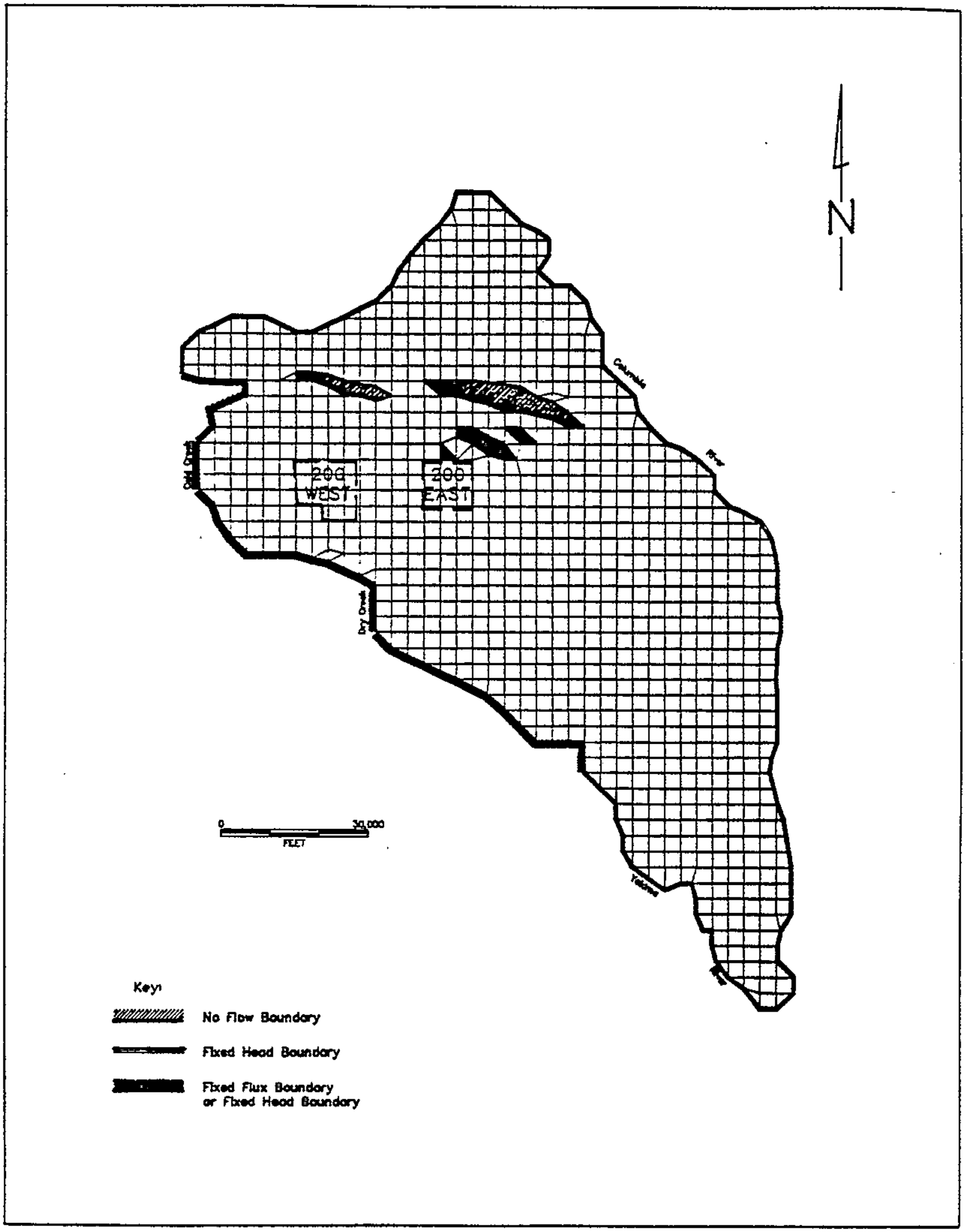

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Figure 1. Finite Element Grid.

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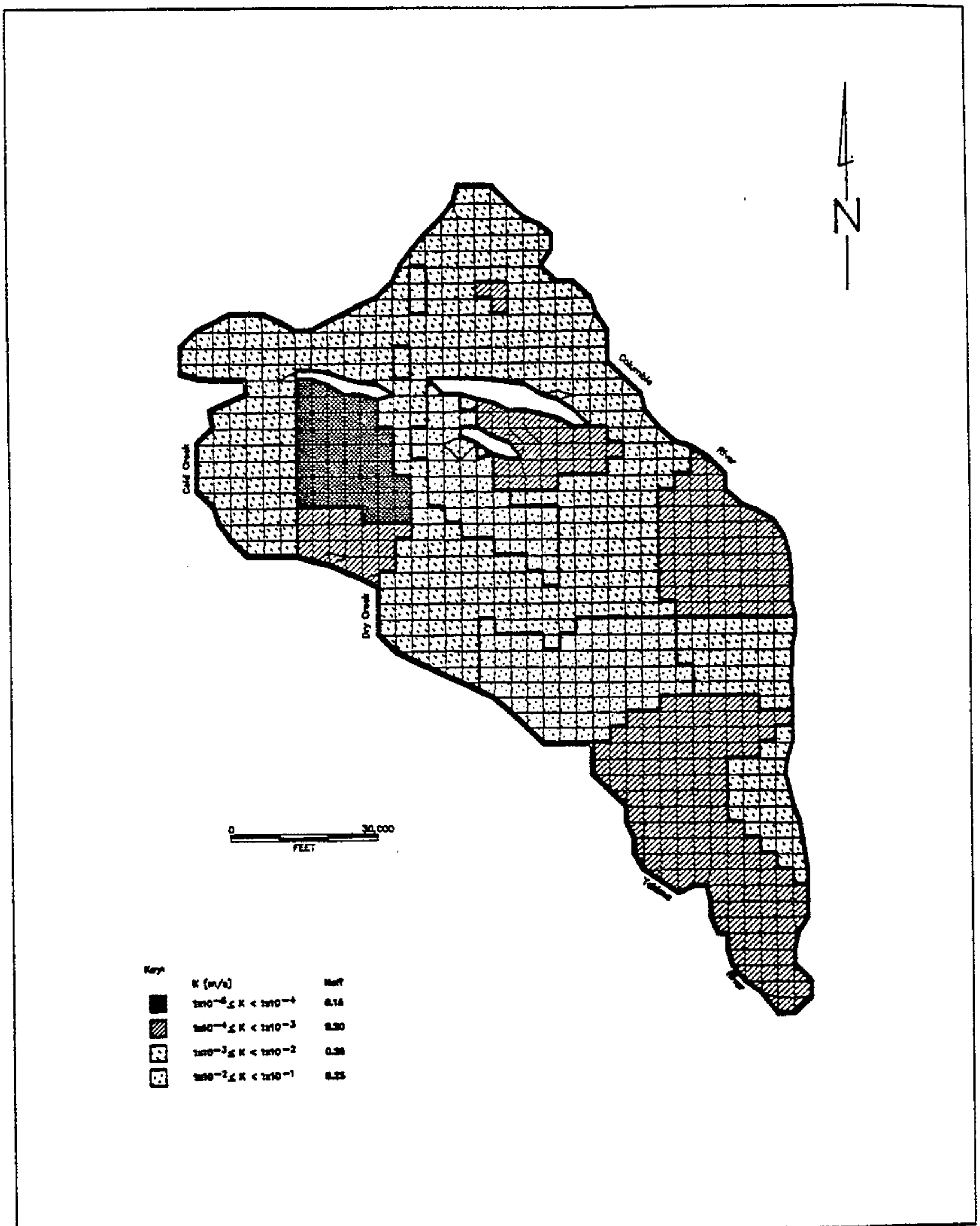

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Figure 2. Model Hydraulic Conductivities.

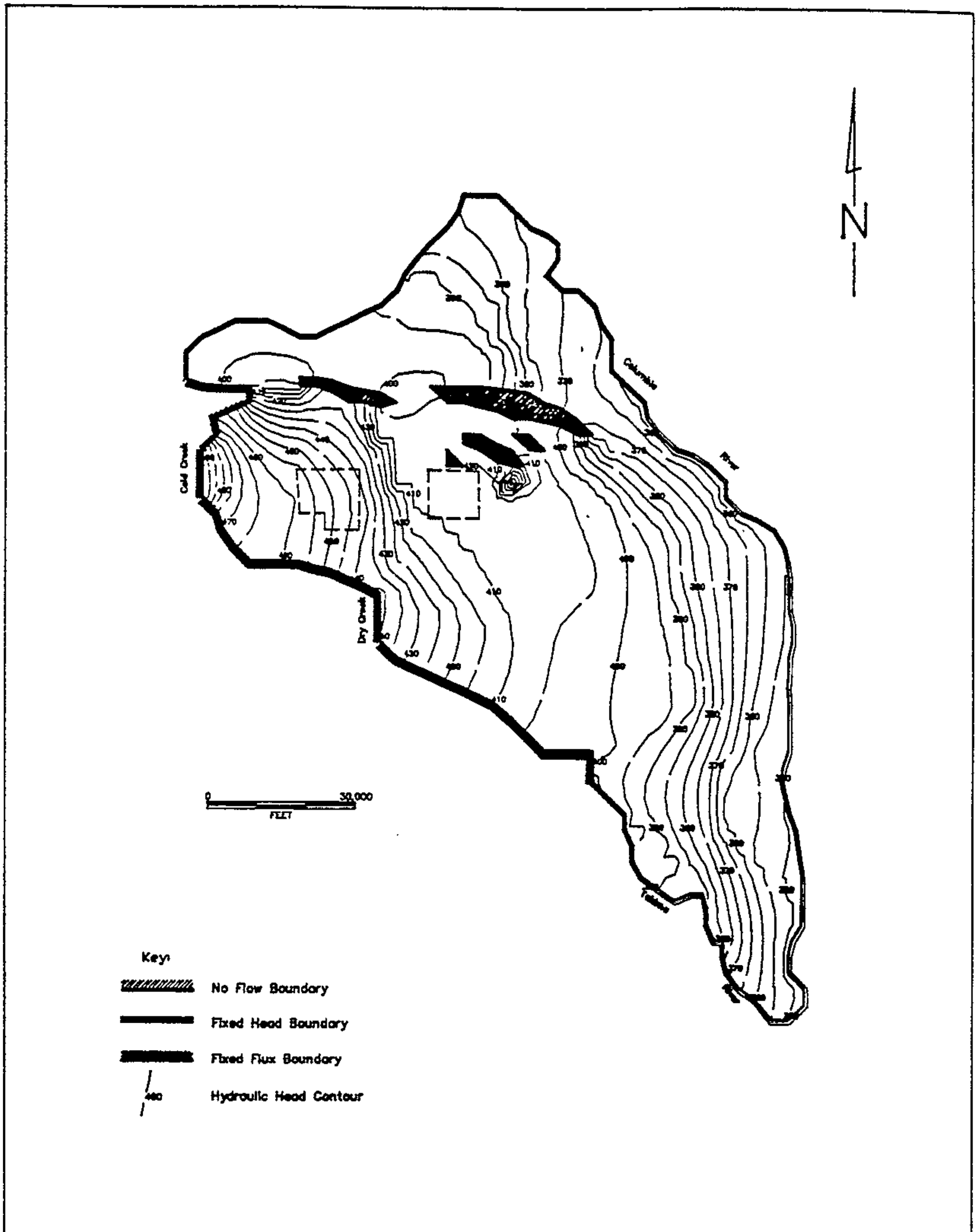

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Figure 3. Hydraulic Head Contours.
Case 1: With B-Pond Operating.

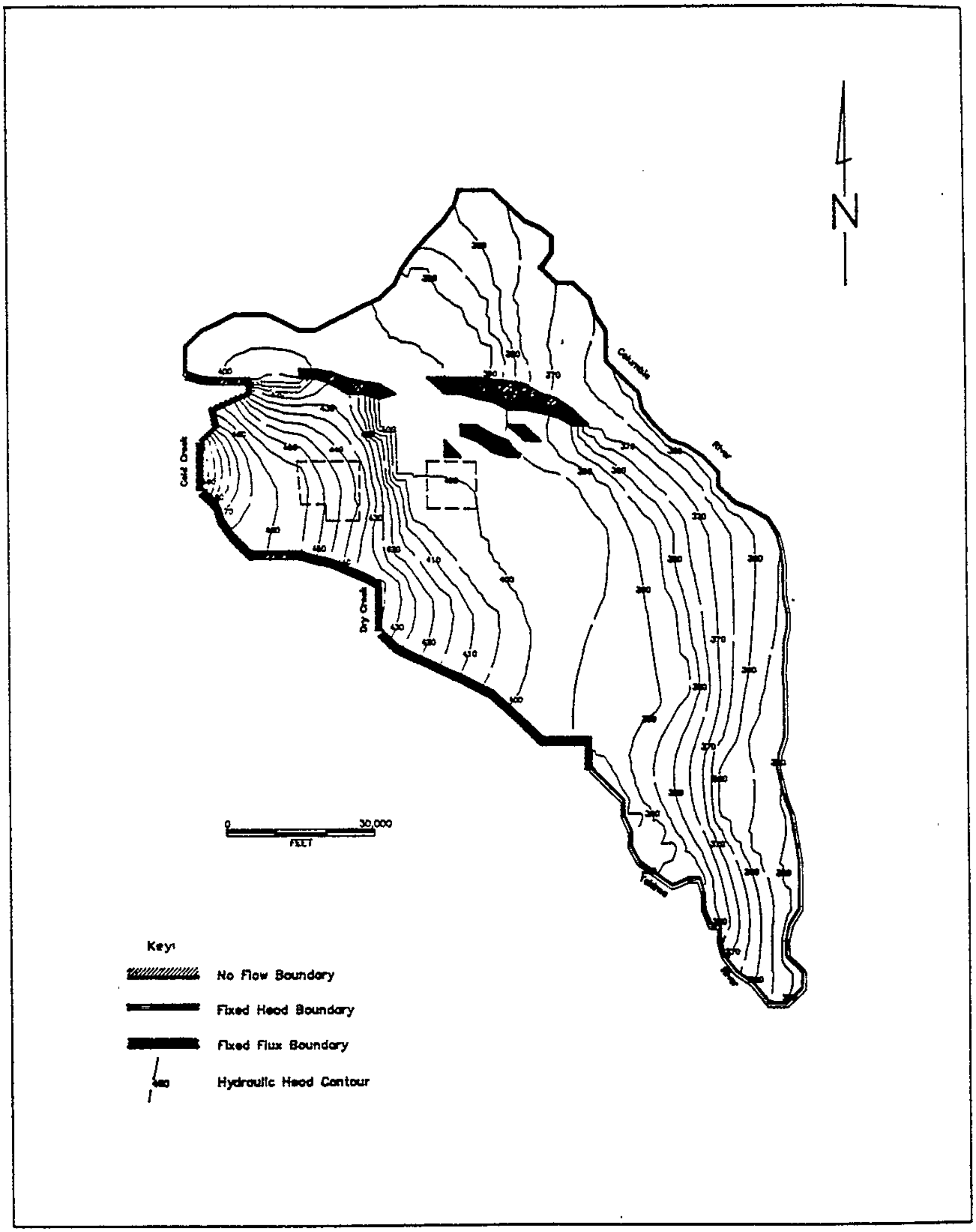

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Figure 4. Hydraulic Head Contours.
Case 2: Without B-Pond Operating.

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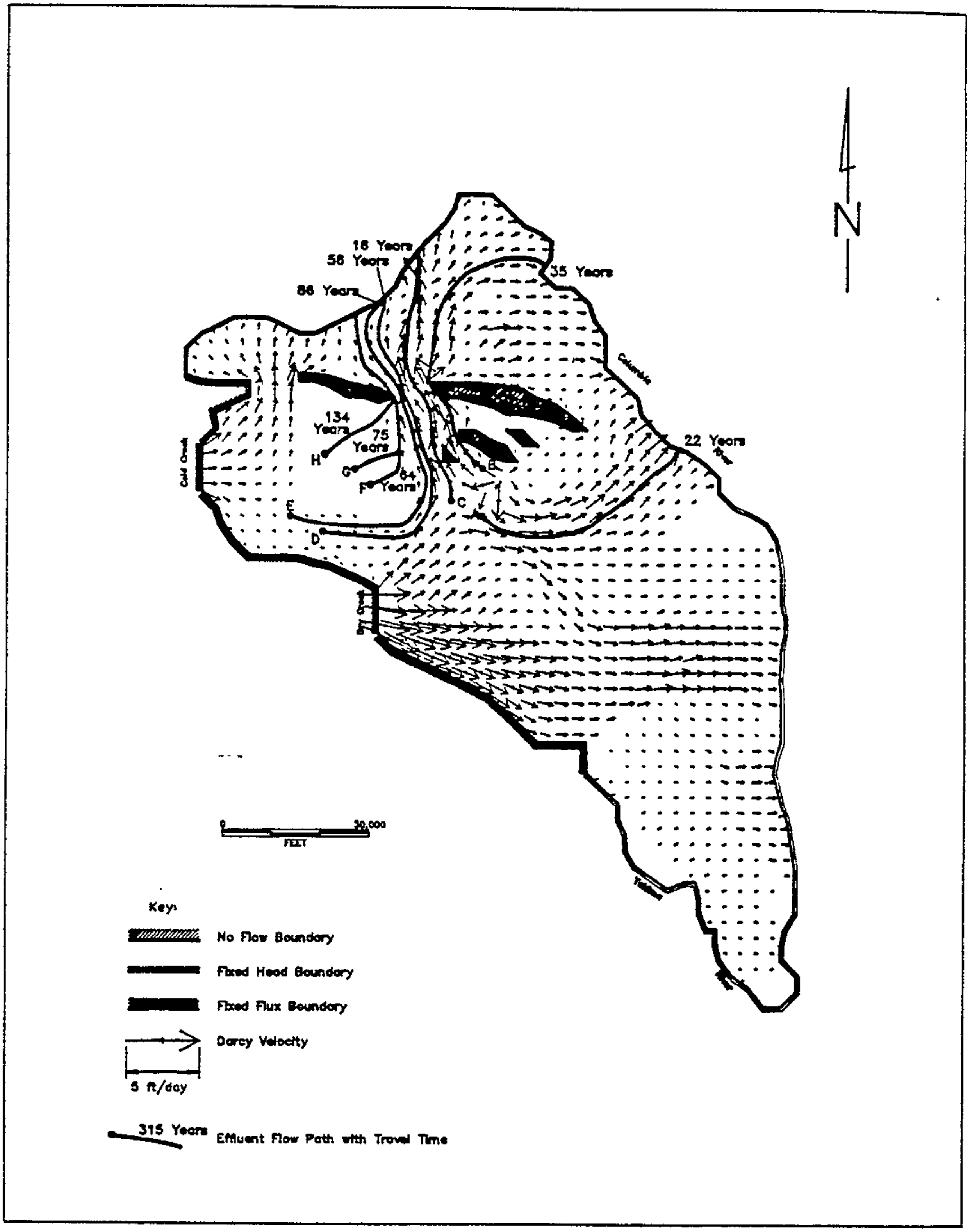

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Figure 5. Alternative Tritium Crib Locations and Flowpaths. Case 1: With B-Pond Operating.

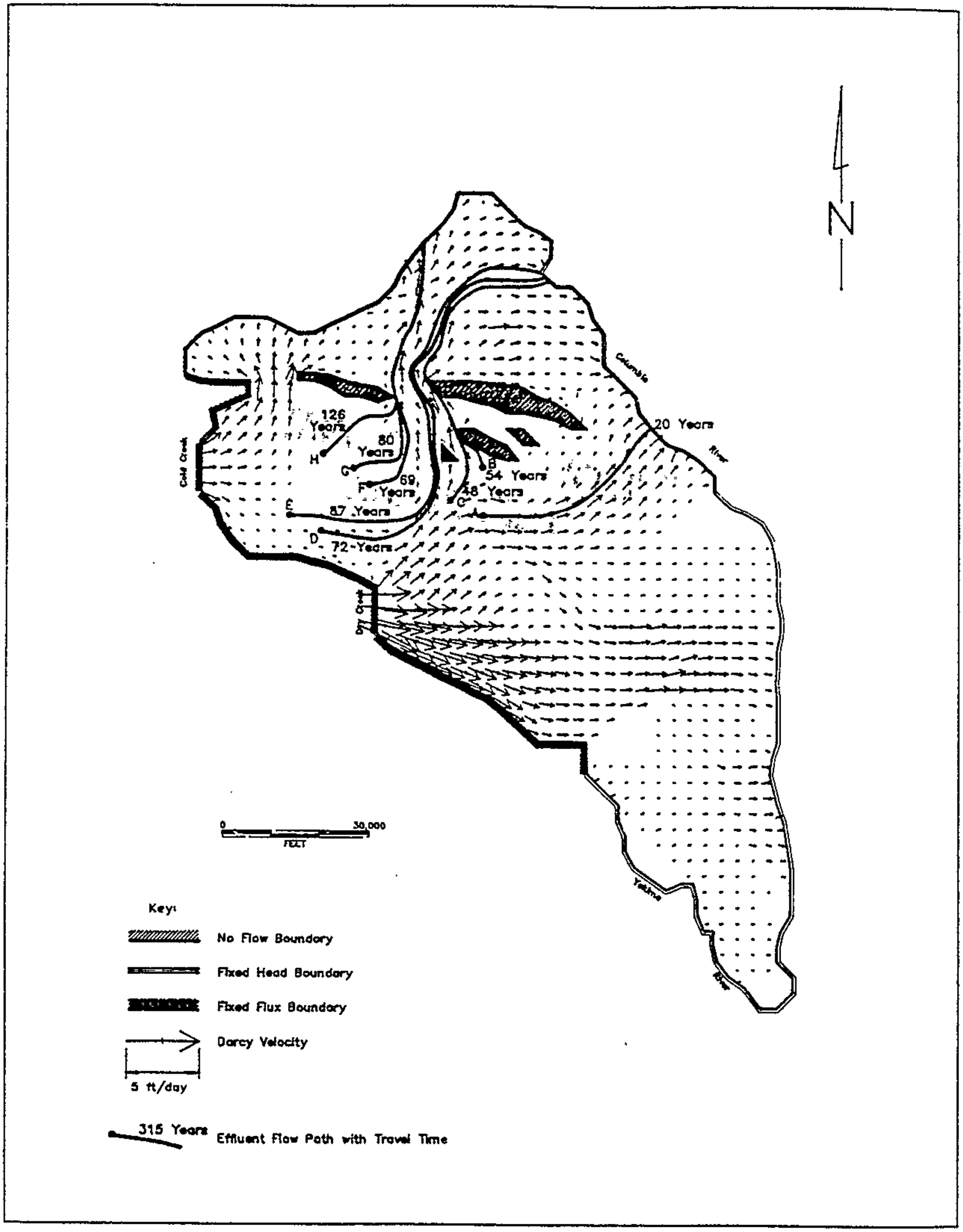

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Figure 6. Alternative Tritium Crib Locations and Flowpaths. Case 2: Without B-Pond Operating.

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APPENDIX A

NUMERICAL GROUNDWATER MODELING

APPENDIX A

NUMERICAL GROUNDWATER MODELING

PURPOSE

Groundwater modeling was performed to support consideration of the soil column disposal option. Specific objectives of the modeling effort included:

1) Demonstrate how groundwater flow patterns would be impacted by various disposal schemes.

2) Provide estimates of travel time under various disposal schemes.

3) Investigate whether it is possible to dispose of the necessary volumes of effluent to the subsurface without causing groundwater mounding which would impact existing soil contamination.

4) Estimate the dilution due to dispersion during subsurface flow to the Columbia River from various disposal sites.

THE COMPUTER CODE

The computer codes used for this modeling effort are primarily parts of the Golder Groundwater Package. The Package includes state-of-the-art finite element computer programs for simulation of groundwater flow and contaminant transport, as well as graphics programs for presentation of the results. Golder Associates Inc. (GAI) has developed the package to simulate a variety of two- and three-dimensional systems. For the purposes of this modeling effort the program AFPM (Aquifer Flow in Porous Media) was utilized. AFPM is designed to simulate groundwater flow through a system of interconnected aquifers, although only one layer was used in this work. The program accommodates variable aquifer properties, a changing phreatic surface, transient boundary conditions, and other characteristics useful for groundwater modeling at the Hanford Site.

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DEVELOPMENT OF THE CONCEPTUAL MODEL

The initial stage of conceptual model development was to define a domain and discretize that domain into a finite element grid. For purposes of modeling large-scale groundwater flow at the Hanford Site, a two-dimensional grid was defined between the basalt ridges on the west side, and the Columbia and Yakima Rivers on the north, east, and south sides of the modeled region. Locations of the basalt ridge boundaries were determined using maps from Gephart et al. (1979) and Serkowski et al. (1988); the river boundaries were located using the United States Geological Survey (USGS) 7.5' topographic quadrangles. Arbitrary boundaries were defined across Cold Creek and Dry Creek Valleys. The modeled domain along with the nodes and elements comprising the grid are shown in Figure H.1. 976 nodes and 920 elements were used to discretize the domain. Most of the elements were square with side lengths of 3275 feet.

After discretizing the domain, the boundary conditions were defined. Fixed-head conditions were established along the river boundaries using values of head from the June 1987 water table map in Serkowski et al. (1988). For calibration purposes, fixed head conditions were also used across the Cold Creek and Dry Creek Valleys. The head values across these boundaries were fixed according to the observed heads reported on the respective calibration standards discussed in the following paragraphs. Along boundaries defined by basalt extending above the water table the model assumed zero flux conditions across the boundary. Zero flux conditions were also assumed along the base of the aquifer. The validity of these boundary conditions will be discussed in the next section.

Initial hydraulic conductivity values were estimated from Plate III-5 in Gephart et al. (1979). The domain was divided up into 27 regions, each of which was assigned a value for hydraulic conductivity, storativity and specific yield. Storativity and specific yield were only important for transient simulations. Although some transient flow modeling was conducted the results were not found to be relevant to the objectives of the study and are not presented.

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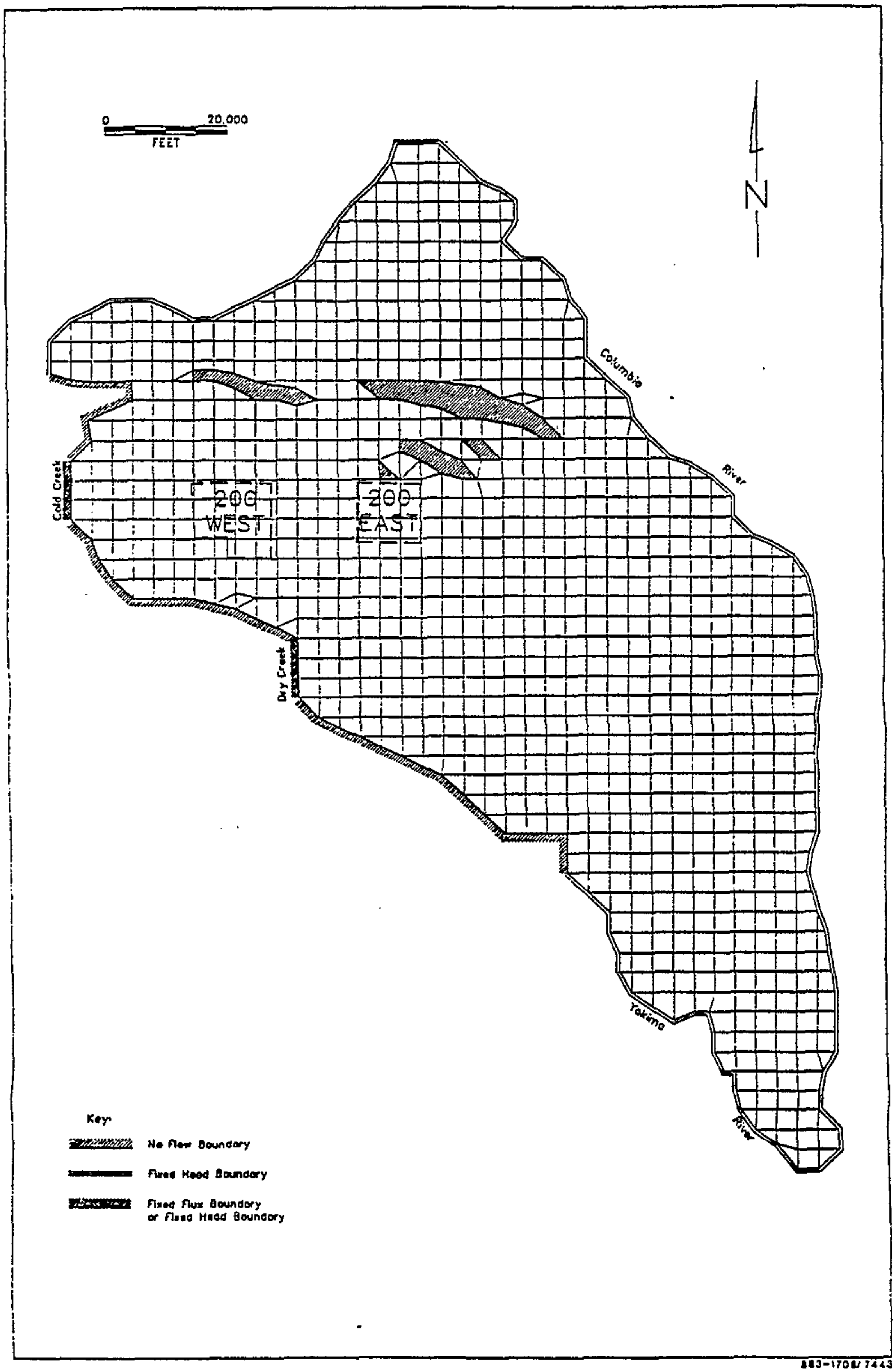

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Figure II-1. Finite Element Grid

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For the first calibration analysis a contour map of 1979 water levels was used as a standard (Plate III-4 in Gephart et al. (1979)). By 1979 the major disposal facilities, B-Pond, Gable Mountain Pond and U-Pond, had been operating for several decades, and groundwater elevations were probably close to steady state levels. The assumed distribution and rates of artificial recharge used for this calibration were estimated from data summarized in Zimmerman et al. (1986); the location and rates of artificial recharge are shown on Figure H.2. Hydraulic conductivities were adjusted until the steady state solution visually approximated the observed 1979 head contours to within about five vertical feet.

To help confirm the estimated hydraulic conductivities a second calibration analysis was performed using a contour map of 1944 water table elevations from Gephart et al. (1979) as a standard. Since effluent discharge was not significant until the mid to late 1940's no artificial recharge was applied to the simulation region. Hydraulic conductivities were adjusted until reasonably close results were obtained for both the 1944 and 1979 calibration standards. When calibration was complete the hydraulic conductivities ranged from 20 to 15000 feet/day. These values are similar to the range of 9 to 10000 feet/day reported by Graham et al. (1981) for the middle Ringold and Hanford units. The hydraulic head contours and Darcy velocity fields for the calibration runs are shown in Figures H.2 through H.5.

During calibration runs fixed head conditions were used across Cold Creek and Dry Creek Valleys. In order to model the various effluent disposal schemes it was necessary to allow the head elevations to change along these sections of boundary. Consequently, these boundaries were changed from fixed head to fixed flux boundaries. The amount of flux across the Cold Creek and Dry Creek boundaries for simulation of future disposal schemes was fixed at the rate which occurred in the 1979 calibration run. These fluxes are as much as ten times larger than those calculated by others (Graham et al. (1981)). Implications of these discrepancies are discussed in the following section.

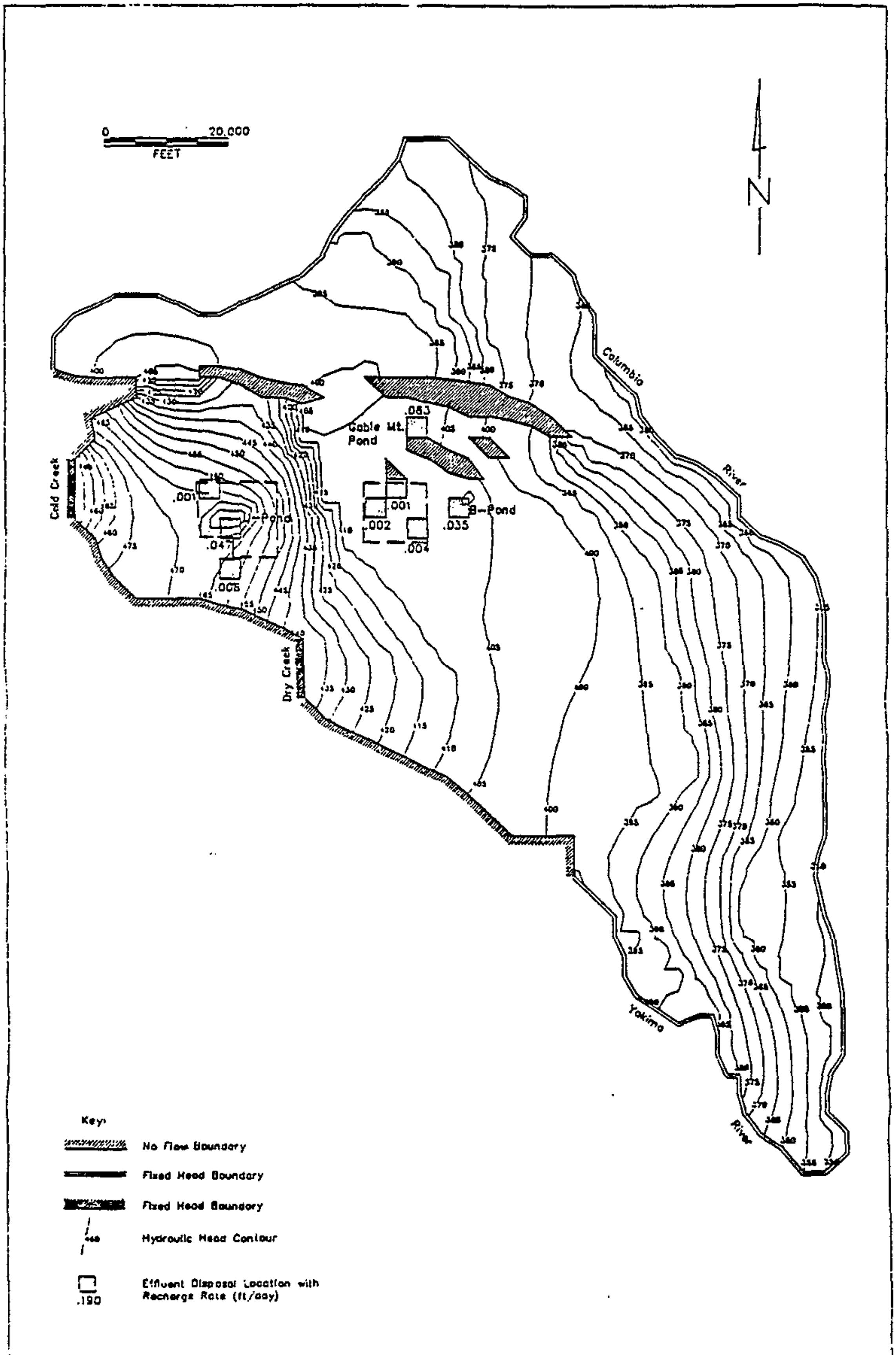

Figure II-2. 1979 Calibration Results Hydraulic Head Contours

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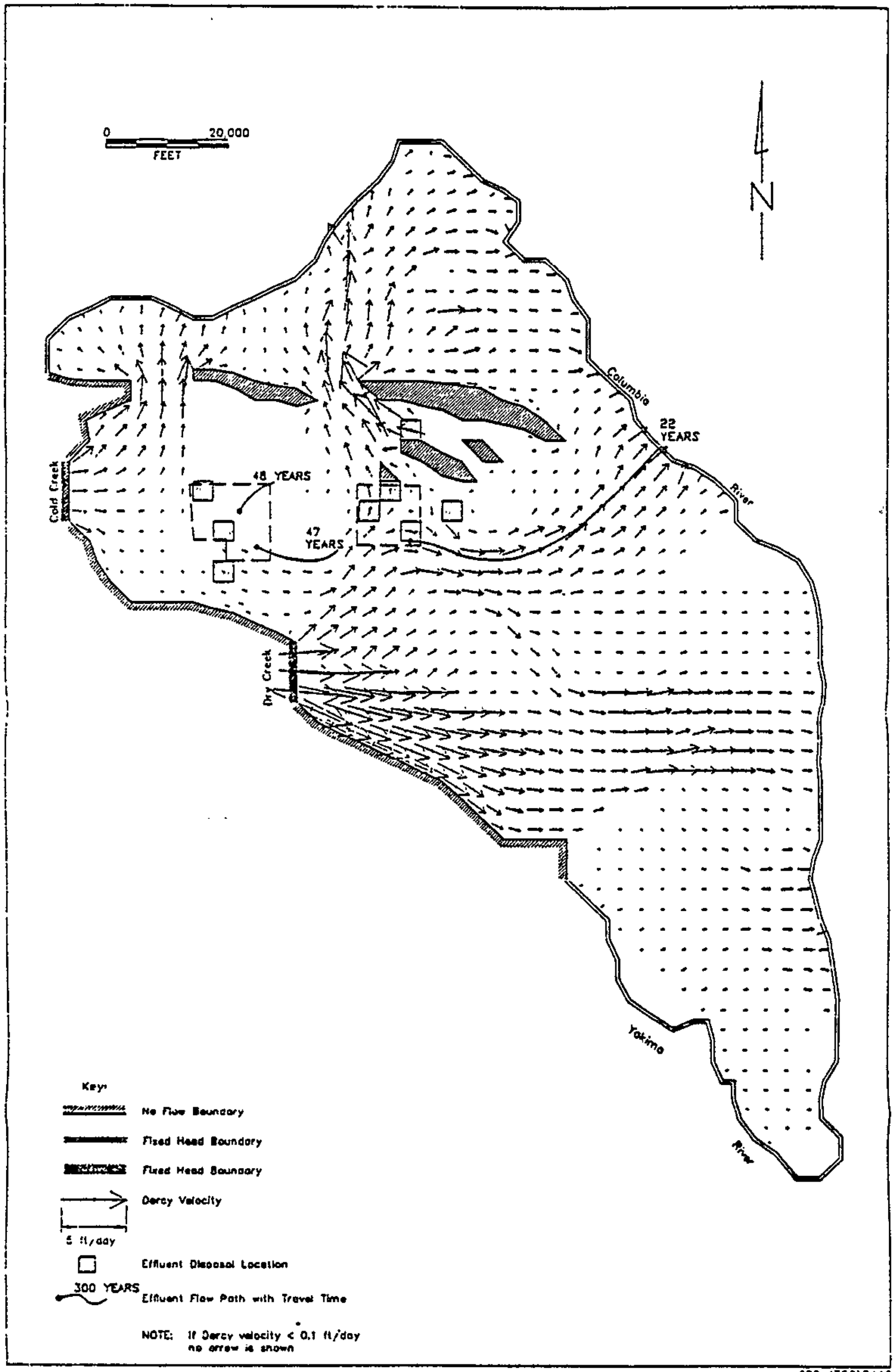

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Figure II-3. 1979 Calibration Results Darcy Velocity Vectors

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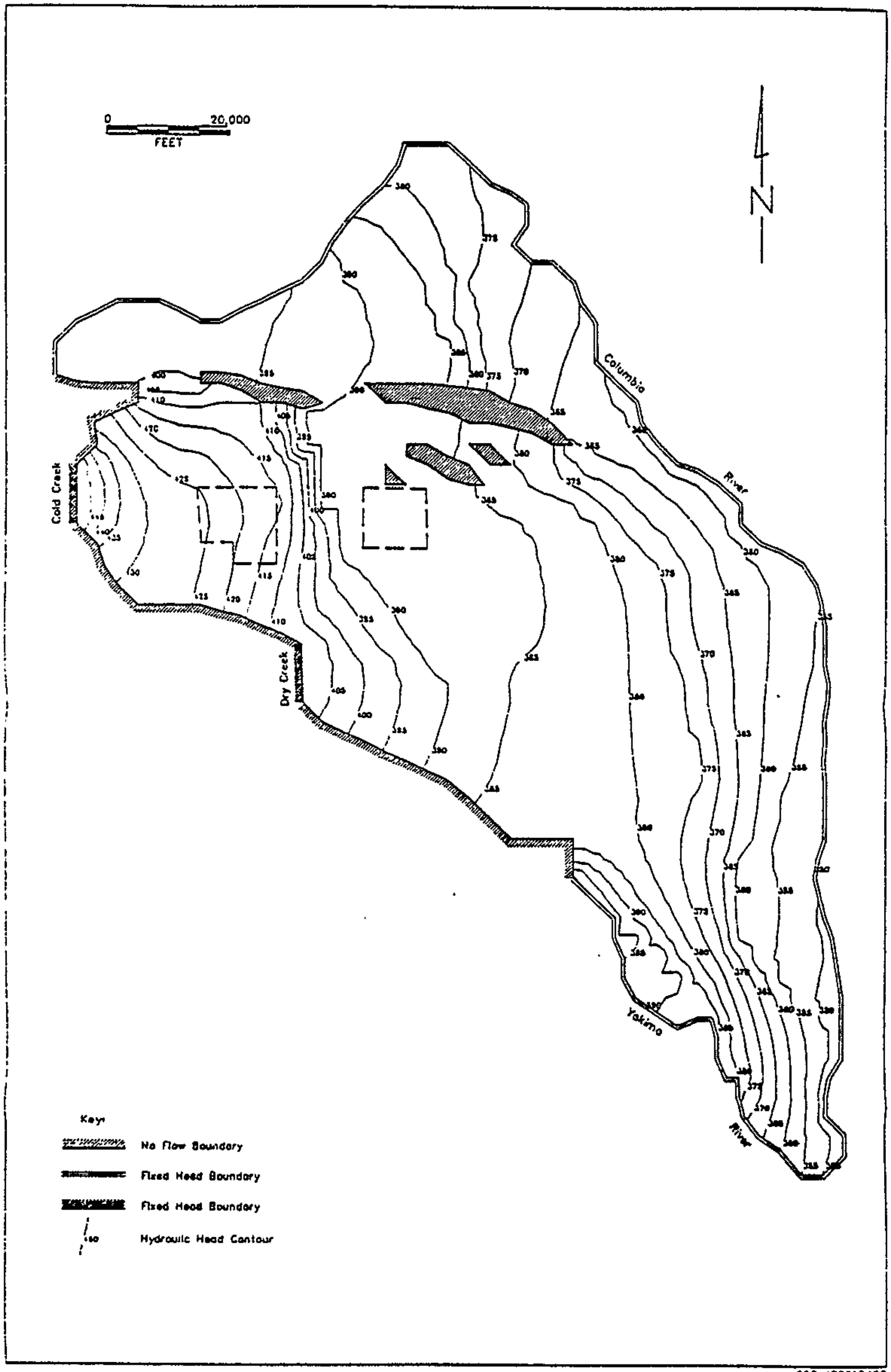

Figure H-4. 1944 Calibration Results Hydraulic Head Contours

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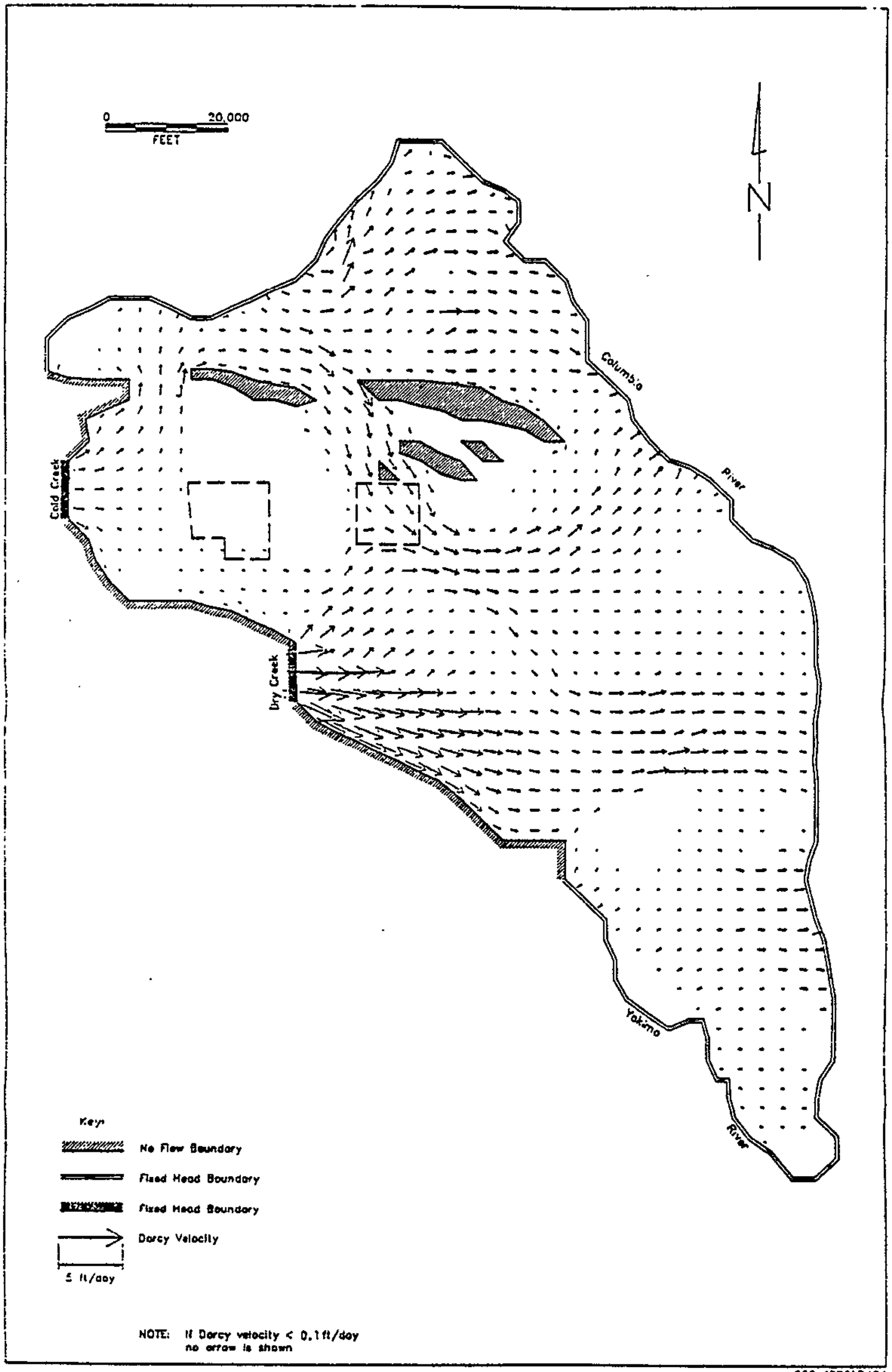

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Figure II-5. 1944 Calibration Results Velocity Vectors

ASSUMPTIONS AND LIMITATIONS

Any modeling of groundwater processes requires some assumptions. An explanation of the rational for the assumptions is helpful for assessing the uncertainty of the results. The assumptions used in this modeling effort are discussed below.

1) Fixed head boundary conditions were used along the Columbia and Yakima Rivers. As explained above, the values of head were fixed at elevations reported in a map of 1987 water levels. If a low permeability layer exists along the base of the river a fixed head boundary condition may not be the most appropriate. Since the naturé of any low permeability layer is presently unknown, we decided to use fixed head conditions. Furthermore, any fluctuation in the stage of the river may cause transient changes in groundwater flow not accounted for in this conceptual model. These effects should be confined to the region near the river and were not important to the objectives of this modeling effort.

2) Zero flux conditions were assumed along the basalt ridge boundaries and the base of the aquifer. Although flow probably occurs across these boundaries, quantifying this flow is virtually impossible given the current state of knowledge.

3) Natural recharge due to infiltration of precipitation was assumed to equal zero. Lysimeter studies discussed in Gee and Heller (1985) and Gee (1987) have indicated that evapotranspiration removes all precipitation from the soil column if the surface is vegetated. It has also been observed, however, that significant recharge may occur in gravelly surfaces with no vegetation (Gee (1987)). Observations of the Hanford site indicate that vegetation covers most of the surface, suggesting that natural recharge would be insignificant.

   As mentioned in the previous section, our model estimates of fluxes out of Cold Creek and Dry Creek Valleys are considerably higher than those estimated by others. If some natural recharge across the Hanford Site were allowed, due to precipitation or due to flux across no-flow basalt boundaries, the amount of flow from Cold Creek and Dry Creek Valleys required for proper calibration would be lower. In order to achieve this lower flow in the model the hydraulic conductivities near these valleys would have to be reduced; reduction of hydraulic conductivities near these boundaries might impact hydraulic conductivities, groundwater flow patterns, and calculated travel times over the entire site. Because of the calibration approach used in this study, however, the possible changes in site-wide conductivities would not be expected to be large. The reason for this is that the heights of the groundwater mounds and the fluxes that created these mounds were used in the 1979 calibration run to establish the values of conductivity near the mounds. Because the relative values of conductivity were known over the entire simulation region from calibration to head data, knowing the conductivity at the mounds permitted the remaining conductivity values to be quantitatively determined.

4) The thickness of the aquifer was estimated from Plate III-2 in Gephart et al. (1979). Although the base of the aquifer is defined as the top of the uppermost basalt flow over most of the simulation region, the lower Ringold is defined as the base of the aquifer where it is present. The lower Ringold is a low permeability layer which only occurs in the western part of the modeled region (Tallman et al. (1979)). A high conductivity layer, the basal Ringold, is present beneath the lower Ringold. It is possible that flow through the lower Ringold into the basal Ringold may impact groundwater flow dynamics above the lower Ringold. Although the Golder Groundwater Package is capable of modeling multi-layered aquifers, the general objectives of this modeling effort did not warrant the additional time and expense of modeling a second layer.

5) The fundamental flow equations used by the AFPM program are derived using standard assumptions for two-dimensional flow modeling, including no vertical flow, vertical averaging of hydraulic conductivity, and deterministic approximation of the flow parameters. These assumptions, plus the assumption of an isotropic medium, were used in the model. Furthermore, the aquifer was modeled as a phreatic aquifer with variable saturated thickness.

RELIABILITY OF THE MODEL

Given the assumptions discussed in the previous section the reliability of the results is difficult to assess. Rigorous quantification of uncertainty would require extensive sensitivity analysis and/or a stochastic approach which were not warranted considering the objectives of this study. A simple method to evaluate the validity of a model is to compare observed travel times with those predicted by the model. A map of the Hanford Site showing tritium concentrations is presented in Figure H.6. At least three tritium plumes originate from sources in the separations area. One of these plumes originates from the 200 East Area and the other two from the 200 West Area.

The plume from the southeast corner of the 200 East Area includes an elevated pulse of tritium which reached the Columbia River in the mid-1980's (Law and Allen (1984); Serkowski et al. (1988)). Tritium is contained in effluent from the PUREX plant which commenced major disposal to cribs in the southeast corner of the 200 East area in the late 1950's (Zimmerman et al. (1986)). Assuming that the main plume of tritium reached the Columbia between 1983 and 1987, the observed travel time to the Columbia River would be approximately 25 years. In a review of travel time estimates, Freshley et al. (1988) concluded that travel times from the 200 East Area could range from 13-23 years. Using the 1979 calibration results, and a porosity of 0.25, the travel time from the southeast corner of the 200 East area is estimated at 22.5 years. The modeled travel path is shown on Figure H.3. Since travel time varies linearly with the value of porosity used in the calculation, and

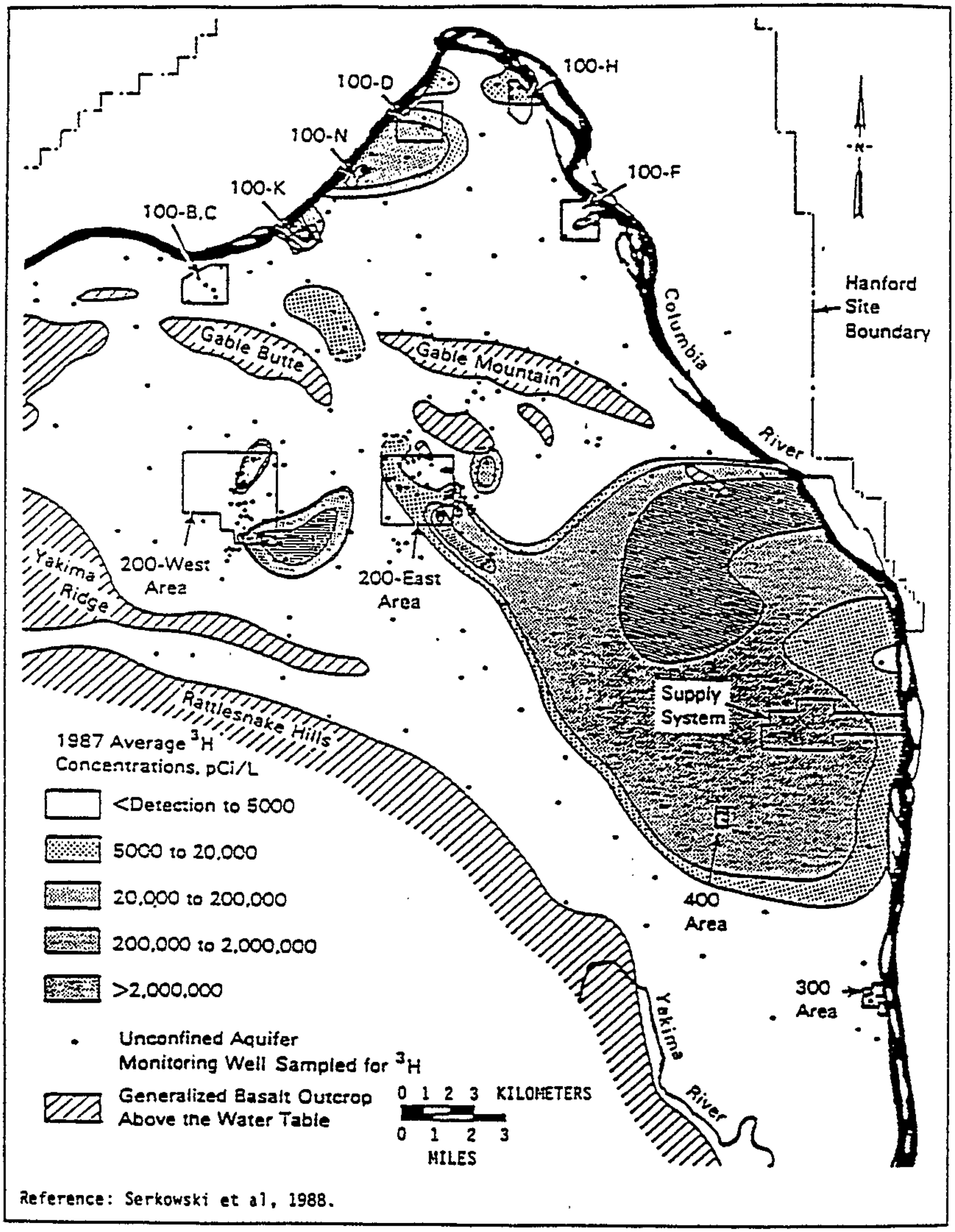

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Figure H-6. Tritium Plume Map for the Hanford Site, 1987 porosity for high permeability materials could range from 0.2 to 0.3, the estimated travel time is probably between 18 and 27 years. This is in agreement with the observed travel time of 25 years.

Two tritium plumes with sources in the 200 West Area are also apparent in Figure H.6. Assuming that both plumes have been produced since effluent disposal began in the late 1940's, they are approximately 40 years old. Travel paths and travel times using the 1979 simulated flowfield are shown on Figure H.3 for transport similar to the observed plumes. Using a porosity of 0.15, the travel times predicted by the model are 47 to 48 years. For the lower permeability materials in the western part of the Hanford Site porosity could vary from 0.1 to 0.2, suggesting a range in travel time from 32 to 63 years. The observed travel time of 40 years is well within this range. The accuracy of the model predictions of travel time lends confidence to the validity of the model.

STEADY STATE RESULTS

Three steady state simulations are presented below. In all three simulations the flux out of Cold Creek and Dry Creek Valleys is fixed at the rate which occurred in the 1979 calibration run.

Simulation 1 is for the case when no effluent is disposed to the groundwater. The resulting water table map and velocity vectors are shown in Figures H.7 and H.8. As expected, the mounding beneath the 200 West Area and beneath B-Pond has dissipated. The results differ from the 1944 calibration run because the flux out of Cold Creek and Dry Creek Valleys has increased significantly, presumably due to increased irrigation in these valleys.

The other two simulations are for effluent disposal to the subsurface at two different sites. These sites are labeled as the "Primary Disposal Site" on Figures H.9 through H.12. Since one criterion for a subsurface disposal site was to avoid impacting existing vadose zone soil contamination, the locations were chosen to lie well outside known solid or liquid waste disposal sites. In addition, the locations were within the high transmissivity zone running through the

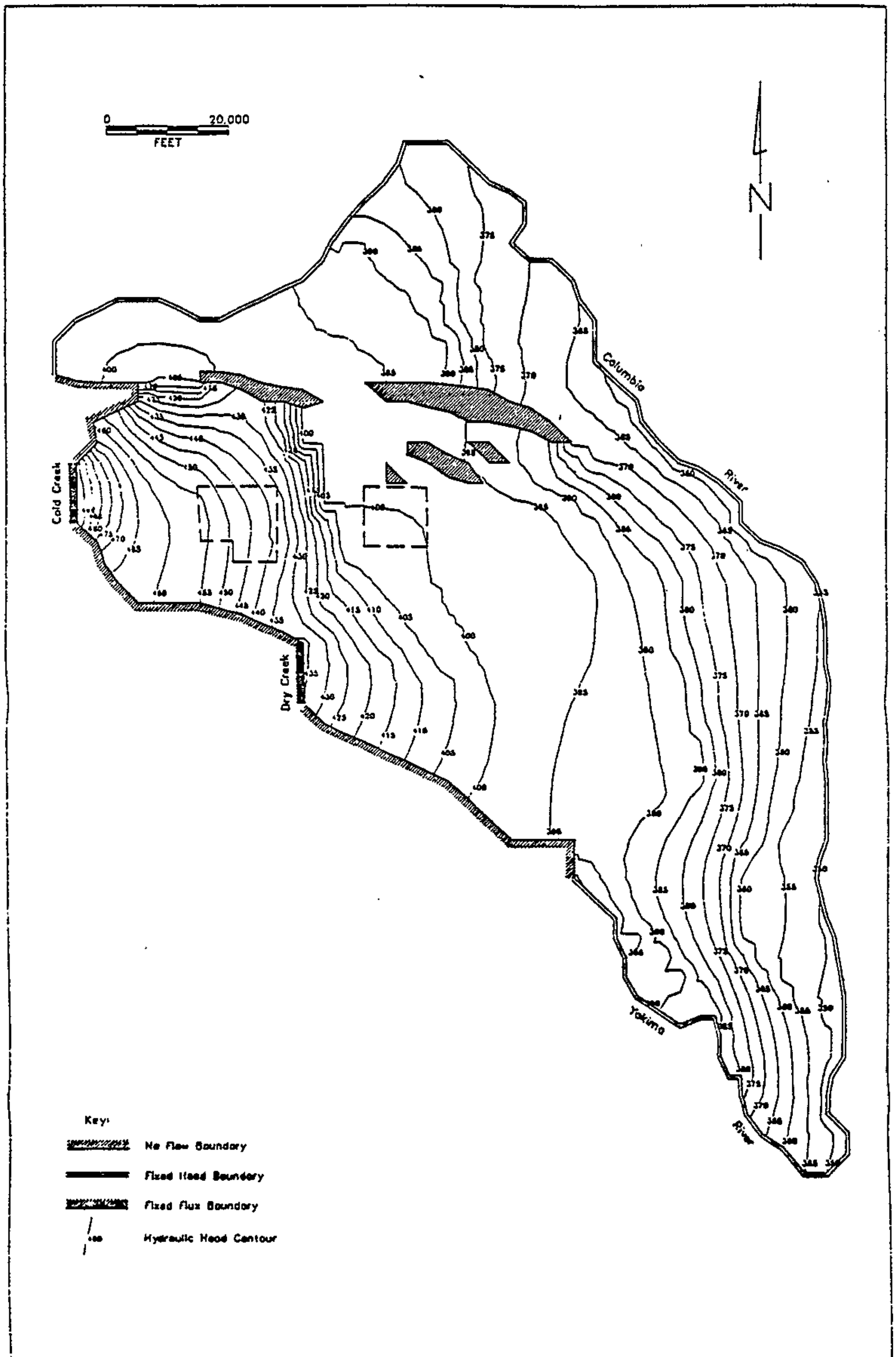

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Figure II-7. Simulation 1 - No Discharge Hydraulic Head Contours

9 2 1 2 6 1 7 0 0 7 3

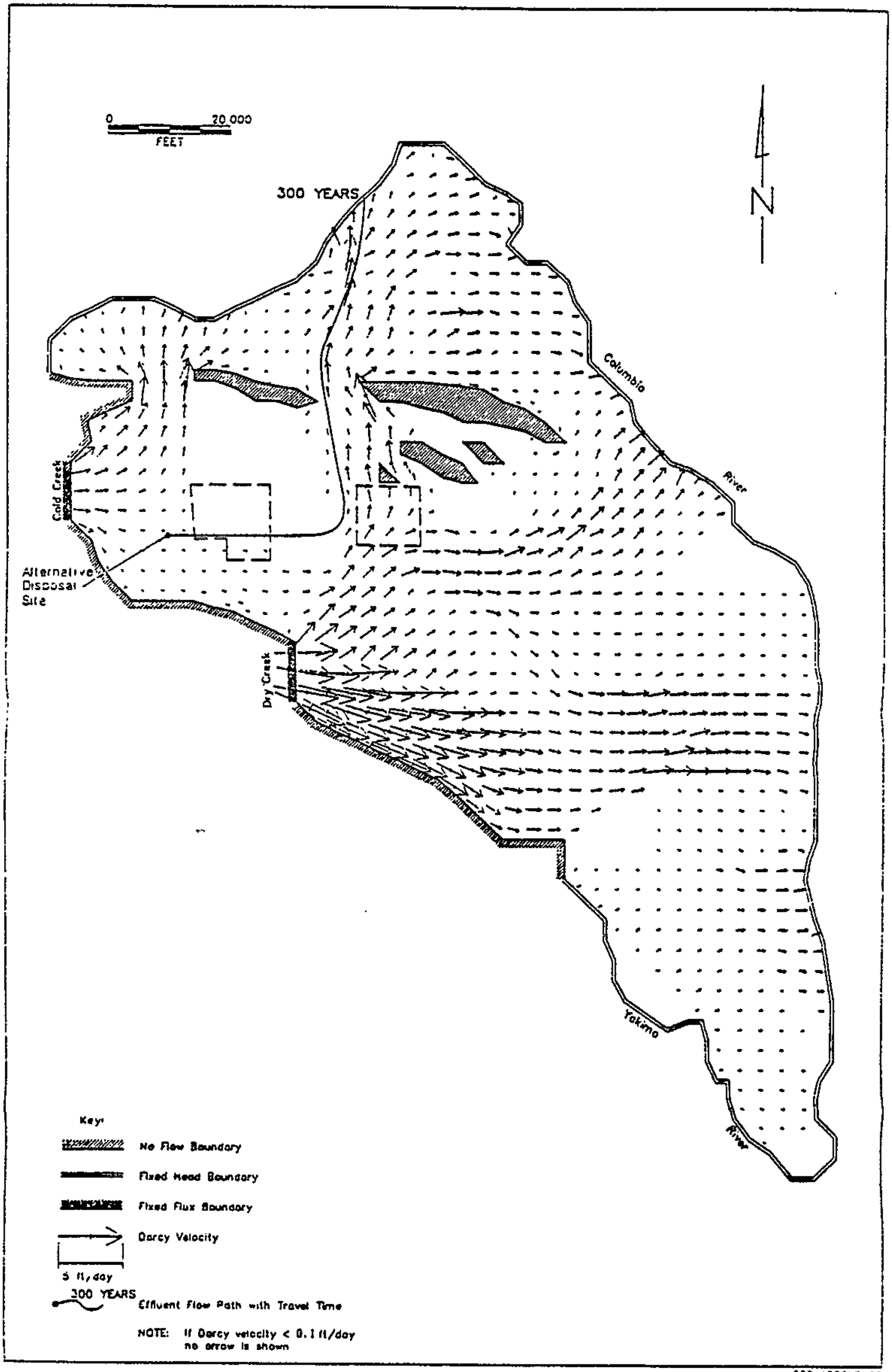

Figure II-8. Simulation 1 - No Discharge Velocity Vectors

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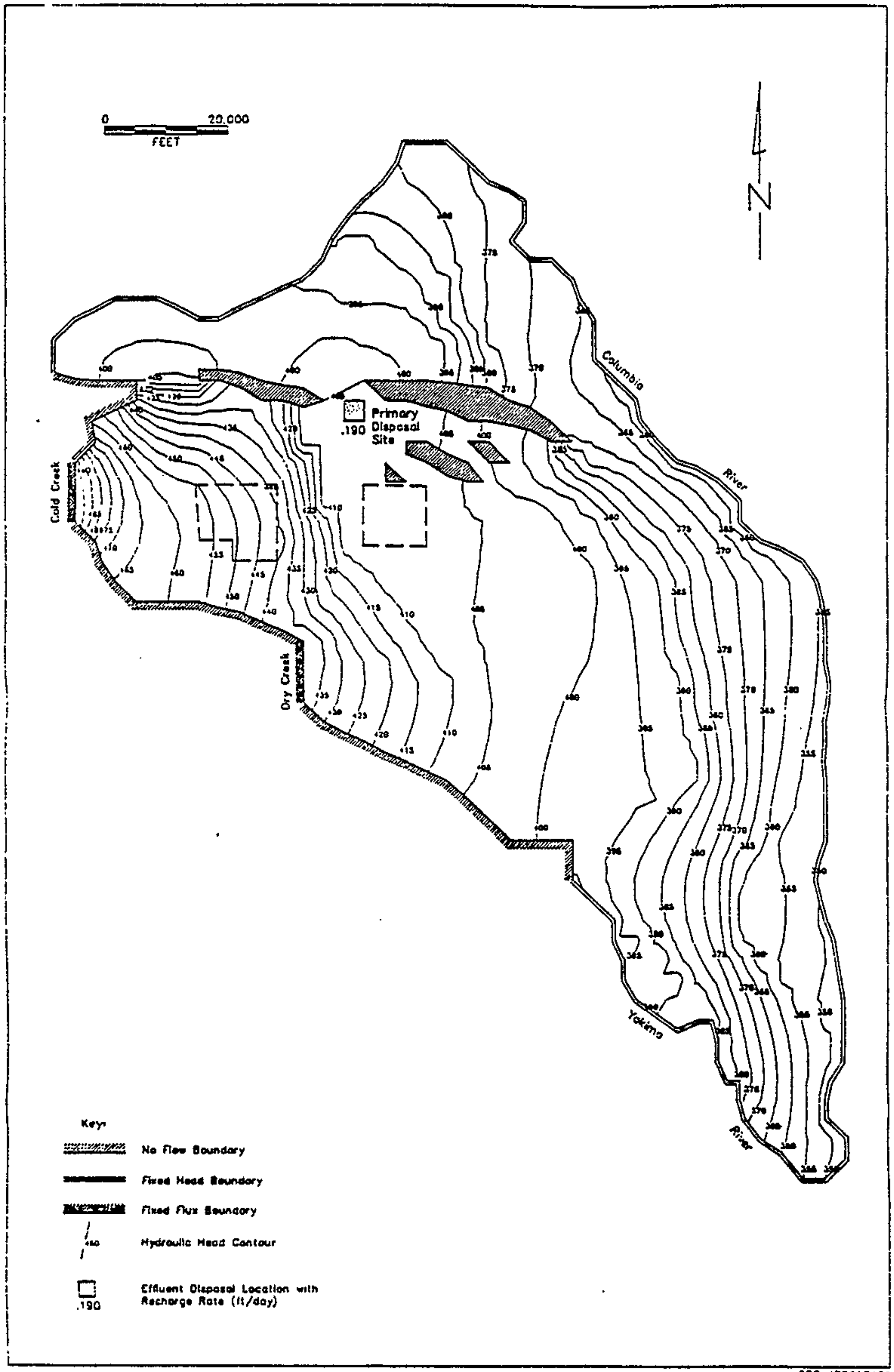

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Figure II-9. Simulation 2
Hydraulic Head Contours

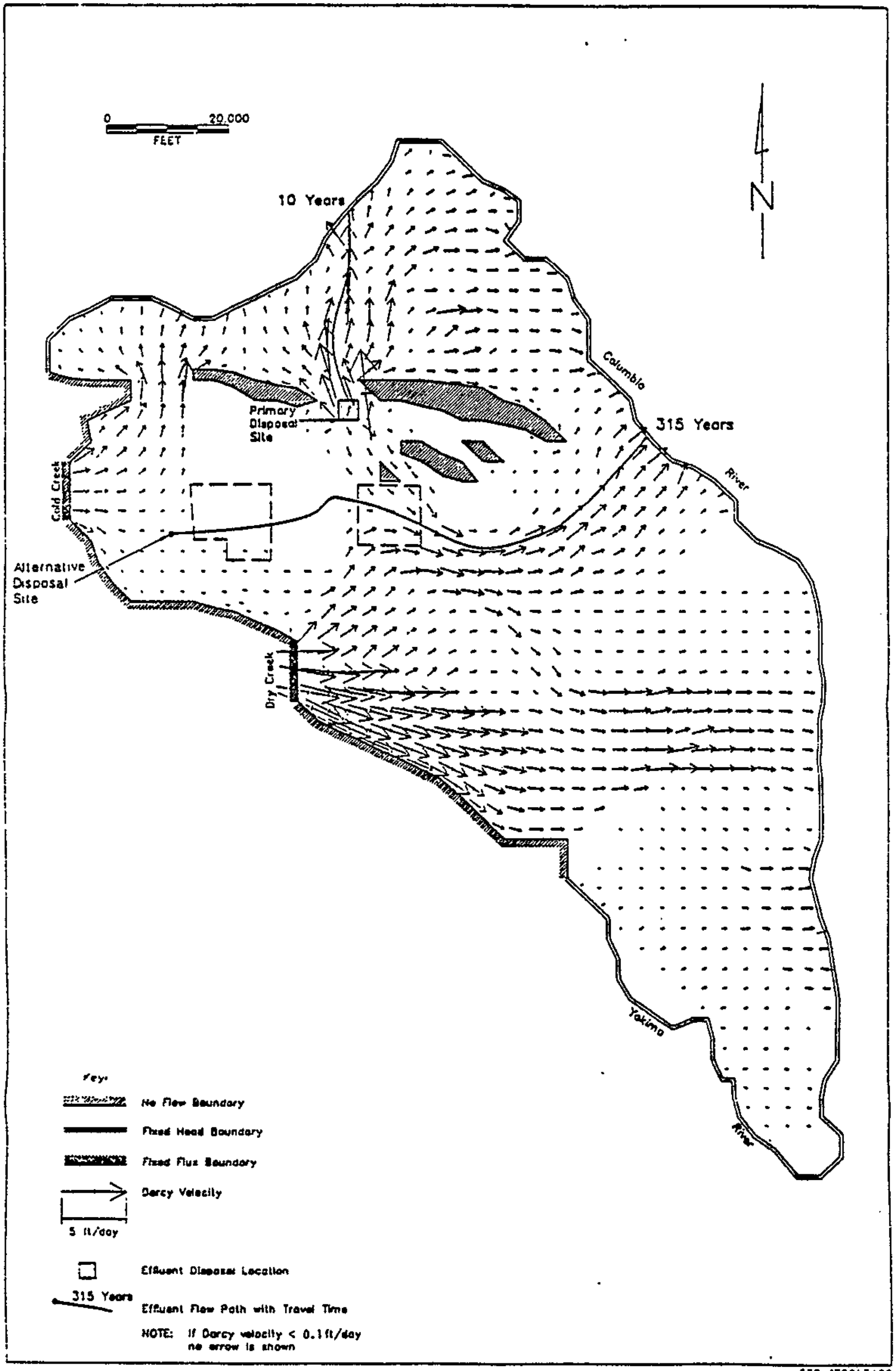

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Figure II-10. Simulation 2
Velocity Vectors

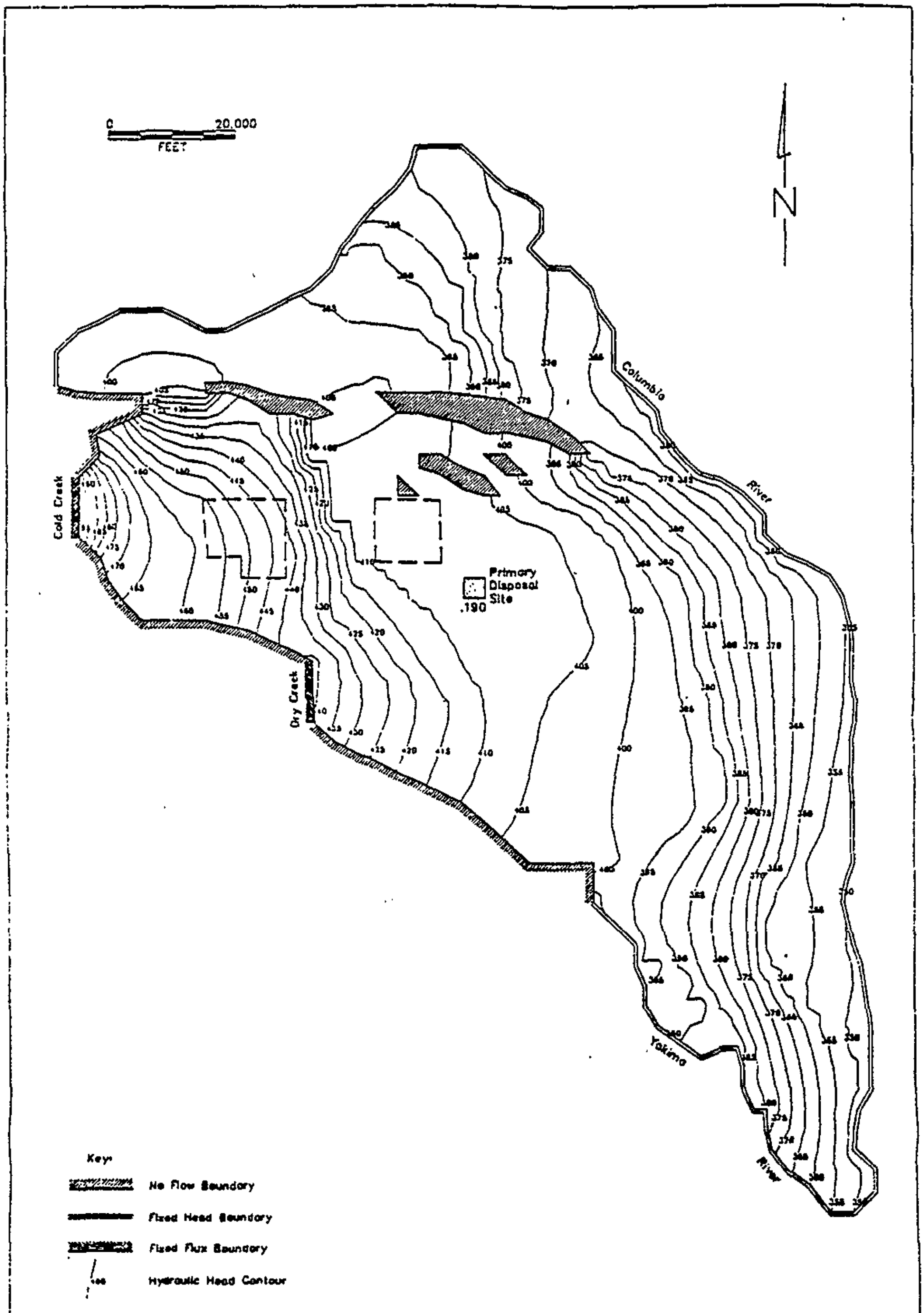

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Figure II-11. Simulation 3
Hydraulic Head Contours

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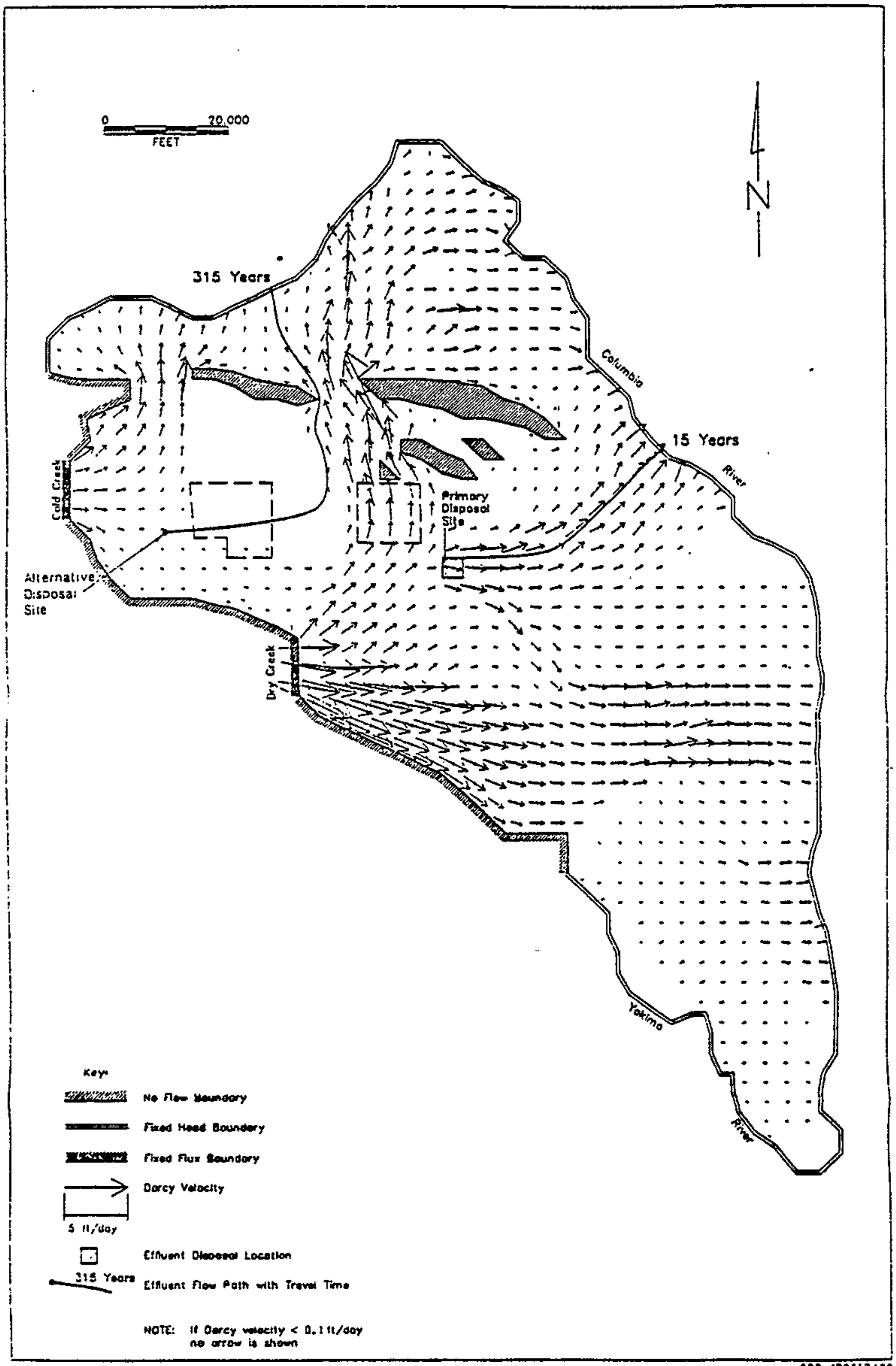

Figure II-12. Simulation 3 Velocity Vectors

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200 East Area to minimize the height of mounding. In both simulations mounding was less than five feet and would not be expected to impact any existing soil contamination. The amount of effluent released in the simulations was 2 million cubic feet per day, approximately equal to the total effluent presently produced at both the 200 East and 200 West areas. As shown in the figures, the recharge has been uniformly distributed over one grid element at a rate of 0.19 feet/day.

Simulation 2 is for a disposal facility located near Gable Mountain Gap, approximately four miles northwest of the proposed retention area at B-Pond. The results are shown in Figures H.9 and H.10. Using a porosity of 0.25, the shortest travel time to the Columbia River from the disposal site is estimated at 10 years. A major disadvantage of this site is that it is located very close to an erosional window through the Rattlesnake Ridge Basalt Flow to the uppermost interbed aquifer (Graham et al. (1984)). Due to the potential for contamination, it would be undesirable to induce flow from the suprabasalt aquifer to a basalt interbed aquifer.

Simulation 3 is for a discharge facility about two miles south of B-Pond. Results are shown in Figures H.11 and H.12. Assuming a porosity of 0.25, the shortest travel time to the Columbia River is estimated at 15 years. This location appears to be better suited than the Gable Mountain location because it is closer to the proposed retention area and it is not close to any erosional windows to the interbed aquifers.

Inspection of the velocity vectors for the three simulations indicates that groundwater flow patterns would be significantly impacted by different effluent disposal schemes. For example, comparison of Figures H.10 and H.12 show that in Simulation 3 the groundwater flow direction across the 200 East Area is completely reversed from that in Simulation 2. Since changes in groundwater flow patterns would affect the movement of any existing contamination plumes, the location of the disposal facility may require re-evaluation of groundwater monitoring networks for regulatory compliance.

ALTERNATIVE DISPOSAL SITE FOR TRITIUM STREAMS

One objective of this study was to investigate the possibility of disposing tritium-contaminated streams in a low conductivity area with a long travel time to the Columbia River. Based upon the flow patterns observed in these simulations an example site was chosen west of the 200 West Area which maximized travel time to the Columbia River. The location is labeled as the "Alternative Disposal Site" on Figures H.8, H.10 and H.12. Since the tritium-contaminated effluent streams are low volume they would not noticeably alter general flow patterns. For transport through the low-conductivity regions near the 200 West Area a porosity of 0.15 was used. A porosity of 0.25 was used for transport through the higher conductivity regions in the central and eastern parts of the Hanford Site. The pathway and travel time from the alternative tritium disposal site for each of the three steady-state simulations presented in the previous section are shown on Figures H.8, H.10, and H.12. For the case when no effluent is disposed to groundwater (Simulation 1) the travel time is about 300 years. The other two simulations, when all the effluent is disposed to groundwater, both have travel times of approximately 315 years. These results suggest that disposal of effluent to groundwater, instead of directly to the river, creates a partial barrier which may slightly retard the movement of upstream plumes.

Given the comparison of observed versus modeled travel times discussed earlier in this appendix, it is conservative to assume that travel times from a low-volume effluent disposal site just west of the 200 West Area are greater than 150 years and less that 400 years. Additional study would be necessary to refine this estimate.

DILUTION FACTORS

Dilution of effluent due to dispersion in groundwater will reduce the concentration of chemical compounds before they reach the Columbia River. The amount of dilution will be affected by a variety of factors, including the amount of wastewater being released, the amount of spreading in the unsaturated zone, the velocity of the groundwater beneath the source, the dispersivity of the soil medium, and the distance from the source to the river. Two approaches have been used to estimate the dilution factor, which is defined as follows:

$$\text{Dilution Factor} = C/CO$$

where C equals the concentration at the river and CO equals the initial concentration.

The first approach is to use empirical evidence from the behavior of existing contaminant plumes to determine the dilution factor. As shown in Figure H.6, the highest concentrations of tritium entering the river from the 200 East Area are between 0.2 and 2.0 microcuries/liter. The source of this tritium is the PUREX Process Condensate stream, which is reported in Appendix A to have a concentration of 30 microcuries/liter. Allowing for 25 years of decay would reduce concentrations by 75 percent to 7.5 microcuries/liter. Assuming a maximum concentration at the river of about 1.0 microcuries/liter the dilution factor is estimated as 0.13.

The second approach is to use an analytical transport model. The model used has been described by Domenico and Robbins (1985). It assumes a strip source of constant concentration, a uniform flow field, constant longitudinal and transverse dispersivity, and zero vertical dispersivity. The dilution factors reported here are intended to approximate steady state conditions at the distances of interest. The necessary parameters include the width of the source, longitudinal and transverse dispersivity, and distance. From a review by Gelhar et al. (1985) of many field scale dispersivity measurements a longitudinal dispersivity of 50 feet and a transverse dispersivity of 5 feet was used. Based upon the dimensions of the plume near the southeast corner of the 200 East Area shown in Figure H.6, the width of the source was set equal to 1000 feet. For the primary disposal sites used in Simulations 2 and 3 the dilution factor is about 0.5. Due to the greater travel distance, the dilution factor for the alternative disposal site is reduced to about 0.35. This analysis indicates that between the primary disposal site and the alternative disposal site the dilution factor is reduced by about one-third.

The dilution factors obtained from the analytical model simulations are higher than those estimated from the empirical evidence. The modeled results are quite sensitive to the width of the source and transverse dispersivity, neither of which are known with much certainty. Furthermore, if vertical dispersion were accounted for in the analytical model the dilution factors would be decreased. Given the uncertainty of the model it is probably advisable to rely more upon the empirically based results.

SUMMARY

To support investigations of the soil disposal option a numerical groundwater model was developed. The model was used to simulate large-scale flow at the Hanford Site. This modeling, supported by field observations and simple analytical modeling, resulted in the following conclusions:

1) Travel times to the Columbia River from two potential disposal sites located in the vicinity of B-Pond was 10 to 15 years. Travel times from an alternative site near the 200 West Area for tritium-bearing streams could range from 150 to 400 years.

2) The dilution factor from proposed disposal sites near the 200 East Area was estimated to be about 0.1 to 0.5. Analytical model results suggest that from the alternative disposal site (west of the 200 West Area) the dilution factor was approximated one-third less than at the primary disposal site (near the 200 East Area).

3) Disposal of proposed effluent streams to the high-transmissivity region running beneath the 200 East Area would probably not create groundwater mounding up into contaminated soil regions.

4) Different disposal schemes will significantly impact groundwater flow patterns and movement of existing contamination plumes.

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